Using GLEAMS and REMM to Estimate Nutrient Movement from a Spray Field and through a Riparian Forest

B. K. Gerwig, K. C. Stone, R. G. Williams, D. W. Watts, J. M. Novak

Abstract. *With the increased number of large animal production facilities in eastern North Carolina, nutrient accumulation is becoming a problem in surface waters and groundwater. To protect these water sources, management practices to reduce nutrient movement or accumulation are being evaluated using computer models. The computer models, Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) and a version of Riparian Ecosystem Management Model (REMM), were used to estimate nitrogen and phosphorus transport of nutrients through a riparian buffer zone from an agricultural field that received swine lagoon effluent. The models simulated annual application rates of effluent equivalent to 500 and 1000 kg N/ha. The GLEAMS model provided the weather data and nutrient concentrations in the soil, sediment, and leachate for input into REMM. Assuming a 1000 kg N/ha loading rate, GLEAMS monthly average NO_3–N leachate concentrations were within 14% of the observed data, and REMM–simulated NO_3–N leachate concentration was within 5% of the observed data. Both models provided an adequate estimation of nitrogen transport through the system. GLEAMS simulations of PO_4–P leachate followed the general trend of observed data. However, there was no apparent response in simulated PO_4–P leachate concentrations for the two loading rates (95 and 190 kg P/ha), indicating a problem in the phosphorus calculations in the model. The REMM–simulated PO_4–P leachate was greater than observed concentrations and was affected by the inputs obtained from GLEAMS. The pre–release version of REMM provided good estimates of the nutrient transport, and with a few improvements, official releases of REMM have the potential to provide better estimates of nutrient movement through the riparian buffer zone.*

Keywords. *Riparian buffers, GLEAMS, REMM, Water quality.*

With the rapid growth of the swine industry in the southeastern U.S., animal waste management must be taken seriously. As the industry grows, larger amounts of swine waste are being stored in lagoons and subsequently applied to field crops as fertilizer. If the effluent is over–applied, groundwater and surface water pollution may occur. To reduce potential pollution of these waters, riparian zones can be used as a buffer between the application fields and waterways. Riparian buffers have been shown to be effective in reducing nitrogen and phosphorus loading of surface waters.

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A riparian forest is a complex ecosystem consisting of soils, vegetation, hydrology, and organisms. These forests can stabilize stream banks, provide wildlife habitat, dissipate water and wind energy, increase sedimentation and hydraulic resistance to flow, and provide long– and short–term nutrient storage (Lowrance et al., 1985; and Schultz et al., 1995). In agriculture, riparian forests typically separate waterways from row crops, pasture, or animal facilities (Lowrance et al., 1985). Schultz et al. (1995) suggested that riparian ecosystems should consist of three zones: a grass strip next to agricultural fields, then a shrub strip, and finally several rows of trees. The grass zone aids in reducing the flow velocity of surface water. The shrub strip and trees aid in soil stability with a permanent root system, increase the bio–diversity and wildlife habitats (Schultz et al., 1995), and capture agricultural non–point source pollutants before entering surface waters and groundwater.

Nitrogen and phosphorus were the non–point nutrients of most concern for surface water and groundwater contamination. Two primary removal pathways of nitrogen were denitrification and storage in woody vegetation. Lowrance et al. (1984) also concluded that denitrification was a significant factor in the removal of nitrogen in a riparian forest. The loss of nitrogen via denitrification could be twice the amount of nitrogen exiting the riparian forest.

Lowrance et al. (1985) found that in the first 10 m of the riparian forest, there was an 8– to 9–fold decrease of the

NO_3–N concentration in the shallow groundwater. In the next 40 m of the forest, there was only an additional 1–mg/L decrease in the NO_3–N concentration. Phosphorus can bind to the soil or be stored in woody vegetation (Lowrance et al., 1985).

In two other riparian zones, Haycock and Pinay (1993) studied the change in groundwater NO_3 concentrations during the winter months. The two riparian zones evaluated were a grass (*Lolium perenne L.*) vegetated zone and a forested poplar (*Populus italica*) riparian zone. Within the first 5 m of the poplar riparian zone, approximately 100% of the applied NO_3 was captured. The grass riparian zone retained approximately 84% of the NO_3 applied. In both riparian zones, increases in groundwater flow rates did not cause a significant change in the width of the maximum retention zone (Haycock and Pinay, 1993).

Lowrance et al. (1983) estimated that 96% of the water moved into a riparian zone as subsurface flow, while less than 20% of the nutrients applied to the upland fields entered the riparian forest through the subsurface flow. Of the nutrients entering the riparian forest, 79% of the nitrogen was in the form of NO_3–N. Only 18% of the NO_3–N exited the riparian forest. This denoted a net removal of 9153 kg NO_3–N from water passing through the 472–ha riparian forest. The amount of land devoted to forest, pasture, and fields affected the nutrient inputs and the filtering capacity of the riparian forest (Lowrance et al., 1983).

Peterjohn and Correll (1984) investigated the movement of nitrogen and phosphorus through a riparian forest. The majority of the nitrogen was removed from the groundwater flow (75%). The riparian forest retained approximately 89% of the nitrogen that entered. The majority of phosphorus (94%) entered the riparian forest through the surface runoff from cropland. The riparian forest retained 80% of the phosphorus that entered. The majority of the total nutrient concentration changes occurred within the first 19 m of the riparian forest (Peterjohn and Correll, 1984).

The Groundwater Loading Effects of Agricultural Management Systems model (GLEAMS) was designed to evaluate the movement of nutrients and pesticides within agricultural management areas. GLEAMS is a continuation of Chemicals Runoff and Erosion from Agricultural Management Systems (CREAMS) (Knisel, 1993). It has a more advanced plant nutrient component, improved climate–soil–management interactions, vertical flux of pesticides, and an improved hydrology program. Three of GLEAMS' main calculation components used in this project were the hydrology, erosion, and nutrient (nitrogen and phosphorus) components. Some field management alternatives were crops grown, fertilizers used, application schedules, and planting and harvesting dates. Most importantly, GLEAMS estimated the non–point source (NPS) pollution caused by a field management plan.

The Riparian Ecosystem Management Model (REMM) was designed to simulate the biological, chemical, and physical processes of a riparian buffer zone. The model has four main calculation components: hydrology, plant growth, nutrient dynamics, and sedimentation and erosion. The model's versatility allows numerous management scenarios for a riparian buffer zone to be evaluated. Some management alternatives that can be investigated are vegetation types, buffer width, and site characteristics. Most importantly, REMM can also estimate the reduction of NPS pollution over time for given site criteria.

The objectives of this project were: 1) to use GLEAMS in combination with REMM to estimate the transport of nutrients from an agricultural field that received swine lagoon effluent and through a riparian buffer zone, and 2) to compare the results to observed data from the system.

Methods and Materials

The project site was a feeder–to–finishing swine farm located in northern Duplin County, North Carolina. The 1600–head swine facility generated approximately 2758 t of waste per year, including 1670 kg of plant–available N per year. The site consisted of the swine facility, a waste application field, and a riparian buffer zone (fig. 1). The 2.4–ha field was planted with Coastal Bermuda grass (*Cynodon dactylon*), which was harvested three times a year. Table 1 describes the characteristics of the swine lagoon effluent, as used in GLEAMS, from Barker et al. (1990) and NCDA Agronomic Division Waste Analysis Reports.

The riparian buffer zone (RBZ) length (to be simulated in REMM) was based on the recommendations of the U.S. Forest Service and the USDA Natural Resources Conservation Service (Altier et al., 1996; Welsch, 1991). The riparian buffer zone was approximately 0.7 ha with an average width of 22 m, a length along the stream of 304.8 m, and an average slope of 4.6%. The riparian buffer zone contained three zones delineated by vegetation management (fig. 2). Zone 1 was a non–managed, undisturbed forest (3 m wide) comprised of Water Oak (*Quercus nigra*) trees along the edge of stream. This zone was intended to protect the integrity of the stream bank. Zone 2 was a managed timber forest (20.5 m wide) constructed of Bald Cypress (*Taxodium distichum*), Red Maple (*Acer rubrum*), Sycamore (*Platanus occidentalis*), and Green Ash (*Fraxinus pennsylvanica*) trees. The trees in

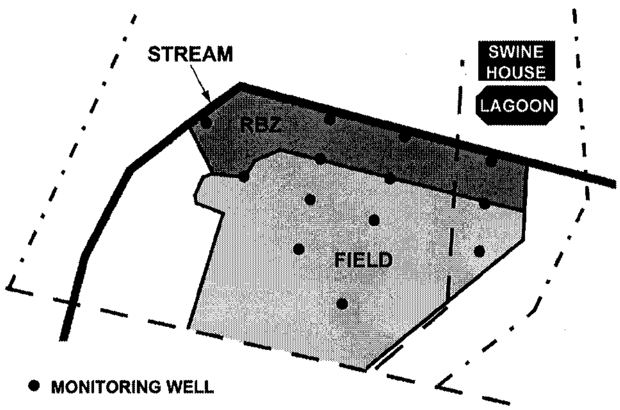


Figure 1. Project site, plan view (not to scale).

Table 1. Characteristics of swine lagoon effluent.

Characteristic	%
Total N	0.0462
Organic N	0.0083
Ammonia	0.0377
Phosphorus	0.0088
Organic P	0.0018
Organic matter	0.1000

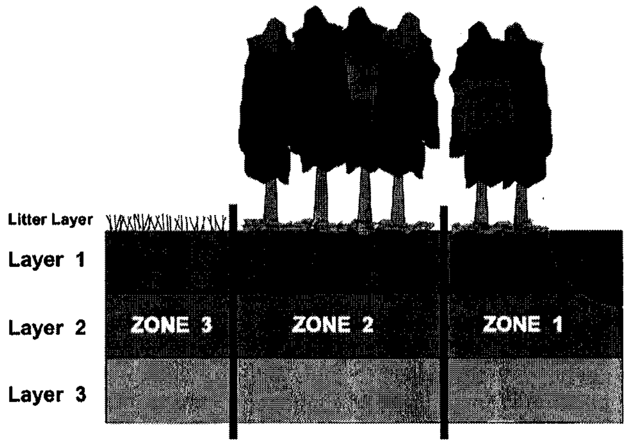


Figure 2. REMM's riparian buffer zone with zones and layers (not to scale).

zone 2 were designed to remove nutrients from runoff and groundwater. Zone 3, a Coastal Bermuda grass strip (3 m wide) at the edge of the field, was designed to slow the runoff flow and increase sedimentation. At the project site, the trees in zones 1 and 2 were planted in April 1993. For simplicity of the simulation, the trees were assumed to be present for the entire simulation period. It was recommended that only one vegetation type be used in each zone. The profile soil was divided into three layers (fig. 2). In REMM, the depth of the water table in each zone determined the bottom of the soil profile. Soil samples and the soil survey provided most of the soil characteristic data.

Weather data were collected for 1990–1997 from several sources and locations near the project site to create a complete data set. Monthly solar radiation from Elizabethtown, North Carolina, (80 km southwest of field site) was provided by the GLEAMS data bank (Knisel, 1993). Dew point temperature, wind velocity, precipitation, and maximum and minimum temperatures were provided by the National Climatic Data Center and the North Carolina State Climate Office. Dew point temperature and wind velocity were an average of data from Raleigh and Wilmington, North Carolina.

Minimum temperature was an average of data from Warsaw and Clinton, North Carolina. Precipitation and maximum temperature were from the Clinton weather station. All of the weather data used in GLEAMS, except precipitation and mean daily temperature, were averaged on a monthly basis from six years of weather data. Daily precipitation and mean daily temperature were used for the simulation period. All weather data for REMM were required on a daily basis and were generated by GLEAMS output for each day of simulation.

The soil at the project site was Autryville fine sand (loamy, thermic, siliceous, Arenic Paleudults). Soil samples from the application field were taken in August 1991 and in March, June, and July 1997. All soil samples were analyzed for total Kjeldahl nitrogen (TKN) and total phosphorus (TP) concentrations and the percentage of sand, silt, and clay in the soil. The Nelson and Sommers (1972) method was used to digest the samples.

Determination of TKN and TP concentrations was accomplished with a Technicon Auto Analyzer (Bran Luebbe, Buffalo Grove, Ill.). TKN analyses followed the automated phenate method. TP analyses followed the automated ascorbic acid reduction method (Greenberg et al., 1992).

Monthly well sample collection began in October 1991 from 18 wells in the application field and the riparian buffer zone. The well samples were analyzed for nitrate–N (NO_3–N), ammonium–N (NH_4–N), total Kjeldahl nitrogen (TKN), ortho–phosphate (PO_4–P), and total phosphorus (TP). Filtered samples were digested with a sulfuric acid digestion using a sulfuric acid:mercuric sulfate:potassium sulfate (100:10:1, w/w/w) catalyst. Determination of nutrient concentrations was accomplished with a TRAACS analyzer (Model 800, Bran Luebbe, Buffalo Grove, Ill.). The EPA methods used for the analysis of NO_3–N, NH_4–N, TKN, PO_4–P, and TP were methods 353.2, 350.1, 351.2, 365.1, and 365.4, respectively (U.S. EPA, 1983). The phosphorus data from November 1993 to August 1994 were considered bad because of changes in sample handling.

GLEAMS was used to evaluate the movement of nutrients within the agricultural management area. Four files were required to run GLEAMS: a hydrology, an erosion, a nutrient, and a precipitation file. The hydrology file contained site parameters, weather data, soil data, and the planting and harvesting schedule. The nutrient file contained the initial nitrogen and phosphorus concentrations in the soil, fertilizer characteristics, and fertilizer application schedule. The erosion file contained Manning's n and soil loss ratios. The precipitation file contained the amounts of rain received daily during the period of simulation (Knisel, 1993). Observed field data were used when available. Other required data were supplied by the GLEAMS database.

The actual application rate of the swine lagoon effluent was unknown until a management plan was developed. Based on previous work (Gerwig, 1998), the application rate was estimated to be between 500 and 1000 kg N ha^{-1} yr^{-1}. Two simulations were developed, each with a different application schedule. Simulation 1 had an application rate of 500 kg N ha^{-1} yr^{-1} (95 kg P ha^{-1} yr^{-1}), not to exceed 1.2 cm per application. Simulation 2 had an application rate of 1000 kg N ha^{-1} yr^{-1} (190 kg P ha^{-1} yr^{-1}), not to exceed 2.4 cm per application. Both simulations had 12 applications of waste per year, with an equal number of applications between each planting and cutting. Three plantings and cuttings were completed between January and October. Although grass is a perennial and was not planted three times a year, GLEAMS required a planting date for growth reference. Simulations were conducted for the period between 1 January 1990 and 31 December 1997. The first two years of simulation were used as a buffer to allow parameters to equilibrate in GLEAMS and REMM and were not used in the data evaluation.

Three soil layers were used in the GLEAMS input files. Table 2 contains the soil properties used in GLEAMS. The soil layers in GLEAMS were determined from the Duplin County soil survey (Goldston et al., 1959). However, GLEAMS further divided these layers into 10 computational layers within the soil profile.

Simulated nutrient concentrations from GLEAMS were compared to the measured nutrient concentrations in soil and well samples from the project site. The well samples from the wells at the edge of the field were averaged and compared to the GLEAMS monthly leachate data leaving the field. The

Table 2. Soil characteristics of the spray field as used in GLEAMS.

Soil Layer	Bottom of Layer (cm)	Porosity (cm^3/cm^3)	FC[α] (cm/cm)	WP[α] (cm/cm)	Silt (%)	Clay (%)	OM[α] (%)
1	66	0.380	0.18	0.03	6.6	2.0	1.0
2	104	0.435	0.18	0.03	8.4	6.5	0.5
3	150	0.380	0.18	0.03	6.2	15.2	0.1

[α] FC is field capacity, WP is wilting point, and OM is organic matter.

nutrient concentrations in the soil were compared through the soil profile at the beginning and end of the simulation period. The 1991 and 1997 field soil data were compared to a five–day average around the dates on which the soil samples were taken.

The results of the GLEAMS simulations were evaluated graphically and mathematically. The GLEAMS output was evaluated for similar nutrient concentration trends with the observed data during the simulation period and within the soil profile. The average monthly nutrient concentration (Y_A) was calculated to aid in the determination of the actual application rate of the swine lagoon effluent. The average monthly nutrient concentration was calculated as:

$$Y_A = \frac{\sum C_M}{n} \tag{1}$$

where C_M is the observed or simulated monthly nutrient concentration and n is the number of months. The percent difference (%Δ) of the total soil and leachate nutrient concentrations from the field edge to the stream was calculated as:

$$\%\Delta = \left(\frac{Z_S - Z_O}{Z_O}\right)100 \tag{2}$$

where Z_S and Z_O are the simulated and observed nutrient concentration in the leachate, respectively.

REMM was used to estimate the nutrient transport within a riparian buffer zone. REMM required four input files to run a simulation successfully. The first file, *.BUF, contained the site geometry, initial nutrient levels, litter and soil properties, and plant types. The second file, *.VEG, contained detailed information about each of the 12 plant types. Much of the detailed vegetation information data was not available for the project site. The third file, *.FIN, contained upland inputs, including surface and subsurface nutrient and water loadings. The fourth file, *.WEA, contained all the weather information common to all zones. The last two files, *.FIN and *.WEA, were constructed almost entirely of user–defined output from GLEAMS. This output provided the following data for these two files: daily precipitation, runoff, percolation, maximum and minimum temperatures, radiation, sediment yield, runoff loss concentrations (NO_3–N, NH_4–N, PO_4–P), sediment loss concentrations (NO_3–N, NH_4–N, PO_4–P), leachate concentrations (NO_3–N, NH_4–N, PO_4–P), and nutrient concentrations by soil layer (NO_3–N, NH_4–N, TKN, TP, PO_4–P) (Altier et al., 1996). Where project site data were unavailable, default data were used from sites in the Coastal Plains near Tifton, Georgia, where REMM is being tested (Bosch et al., 1996; Lowrance et al., 1998; Sheridan et al., 1998). Changes were made where possible to better estimate the project information.

Three soil layers per zone were used in the REMM input files. Table 3 contains the soil properties used in REMM. The data were obtained from the Duplin County soil survey (Goldston et al., 1959) and the soil samples taken within each zone.

REMM's leachate data were compared to the observed data. The REMM output was evaluated for trend similarities with the observed data from 1992 to 1997. For comparison with REMM, wells at the field and stream edges of the riparian zone were compared to the leachate data of zone 1. The nitrate–nitrogen leachate concentrations from REMM and the well samples were compared. The observed PO_4–P concentration in the leachate was compared to the labile phosphorus (LP) leachate concentrations from REMM. There was a linear relationship between PO_4–P and LP in leachate (Paul and Clark, 1989).

The results of the REMM simulations were evaluated graphically and mathematically. The majority of the

Table 3. Soil characteristics for each zone as used in REMM.

Soil Layer	Bottom of Layer (cm)	Porosity (cm^3/cm^3)	FC[α] (cm/cm)	WP[α] (cm/cm)	Sand (%)	Silt (%)	Clay (%)
				Zone 1			
1	9	0.35	0.15	0.09	91.77	7.60	0.63
2	17	0.38	0.15	0.09	91.63	7.39	0.99
3	26	0.40	0.15	0.09	90.77	6.10	3.14
				Zone 2			
1	15	0.35	0.15	0.07	97.13	2.71	0.16
2	46	0.38	0.15	0.07	94.86	4.99	0.15
3	66	0.40	0.15	0.07	95.05	4.38	0.58
				Zone 3			
1	66	0.38	0.18	0.03	91.4	6.6	2.0
2	104	0.435	0.18	0.03	85.1	8.4	6.5
3	150	0.38	0.18	0.03	78.6	6.2	15.2

[α] FC is field capacity and WP is wilting point.

graphical analysis was a comparison of the simulation results within each zone or layer. The Y_A and %Δ were also used to evaluate the REMM data.

Results and Discussion

GLEAMS Simulations

The GLEAMS model was used to simulate nitrogen and phosphorus movement in a swine wastewater spray field. The spray field was simulated at application rates of 500 and 1000 kg N ha^{-1} yr^{-1} (simulations 1 and 2, respectively). When the observed and simulated concentrations were compared, few similarities resulted (fig. 3). As the application rate increased, the simulated peak NO_3–N concentrations increased. After January 1994, the observed NO_3–N concentrations leveled out between simulations 1 and 2.

For a better estimation of the application rate, the average monthly NO_3–N leachate concentrations were compared (table 4). The observed average NO_3–N concentration was between simulations 1 and 2, as expected, but was nearer to simulation 2. The percent difference between the observed average NO_3–N concentration and simulation 1 was 92%. The percent difference between the observed average NO_3–N concentration and simulation 2 was 14%. This indicated that the actual application rate was likely closer to 1000 kg N ha^{-1} yr^{-1}.

The comparison of the 1991 and 1997 simulated TKN soil concentrations with the observed TKN concentrations was in good agreement (fig. 4). In 1991 and 1997, differences between the simulations occurred only within the top 20 cm of the soil profile. This was primarily due to the high TKN concentrations in the litter layer. Below 20 cm, the concentrations for each simulation were almost identical in shape and magnitude. Both simulations estimated decreases

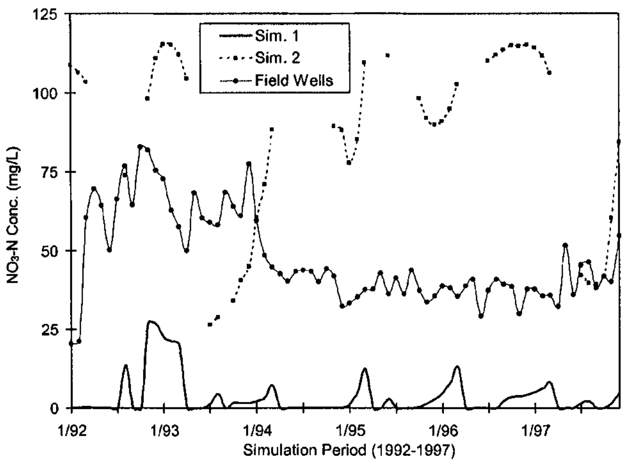


Figure 3. Observed and GLEAMS NO_3–N leachate concentrations during the study period.

Table 4. Average monthly NO_3–N concentrations.

	Y_A (mg/L)	%Δ
GLEAMS Sim. 1	3.5	–92.6
GLEAMS Sim. 2	54.0	14.2
Field wells	47.3	

in concentrations at 66 cm. However, the decrease did not occur at the same depth as the observed data. In the 1997 data comparison, GLEAMS simulated the increase of TKN concentrations in the last soil layer. GLEAMS estimated the trends of the TKN concentrations within the soil profile relatively well.

When the observed and simulated PO_4–P leachate concentrations were compared, the results were not as favorable (fig. 5). Prior to November 1993, there were no similarities between the simulated and the observed concentrations. After August 1994, the observed PO_4–P concentrations and simulated peak concentrations began occurring simultaneously during the later part of the simulation period. GLEAMS estimated that the peak PO_4–P leachate concentrations would remain relatively constant over time. Both simulations overestimated the observed average PO_4–P leachate concentration (table 5).

In 1991 and 1997, the TP in the soil showed the most difference between the simulations within the top 20 cm of the soil profile (fig. 6). In both simulations, TP concentrations remained relatively constant below 20 cm in the soil profile. This was consistent with the observed data. In 1997, the litter layer had a greater effect on the comparison of the simulations and the observed data. Below 20 cm, there was almost no detectable difference between the observed and simulated concentrations.

Comparisons to Previous Work

The trends of the NO_3–N leachate from Hubbard et al. (1987) study were comparable to those simulated by GLEAMS in this project. That study compared the NO_3–N leachate concentrations of fields applied with dairy wastewater at rates similar to simulations 1 and 2. The first similarity was the seasonal variation of the NO_3–N leachate concentrations in both simulations. The simulated NO_3–N leachate concentrations were higher in the winter than in the summer months. Our observed project data did not show seasonal influences in NO_3–N leachate concentrations. Second, over time, there was little or no increase in the NO_3–N leachate concentrations for either application rate. GLEAMS estimated no increase in the NO_3–N leachate concentrations due to the simulation 1 application rate. There was a small increase over time in the NO_3–N concentrations due to the simulation 2 application rate, but observed project data showed a decrease in the NO_3–N leachate concentrations over time. Thus, overall, GLEAMS estimated similar trends of the NO_3–N leachate concentrations as in the study data. This may indicate that the observed data did not follow typical trends.

The trends of a GLEAMS simulation by Yoon et al. (1994) were also similar to those simulated by this project. The Yoon study compared observed and simulated nutrient concentrations from two application rates of poultry litter. The first similarity was the higher nitrogen concentrations within the top 20 cm of the soil profile. The majority of the difference between the two application rates occurred in the top 20 cm. The remaining profile depth had a fairly constant concentration. In addition, the increase in the nitrogen concentrations from the beginning to the end of the simulation period was greater for the higher application rate. The difference in the simulated nitrogen concentrations between the applications rates became more significant over time. Both this project and the Yoon study simulations

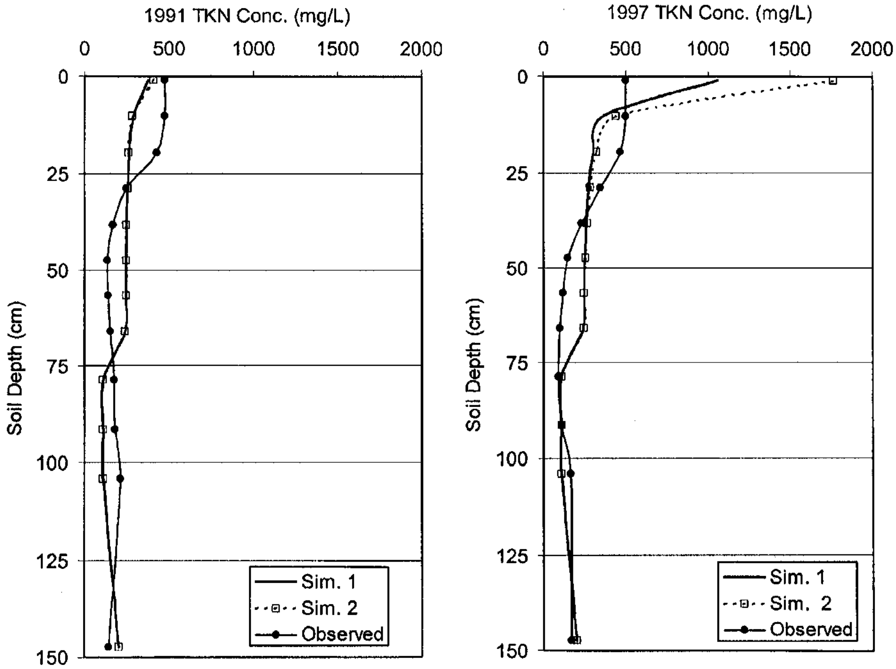


Figure 4. Observed and GLEAMS simulated TKN soil concentrations versus soil depth for 1991 and 1997.

resulted in an underestimation of the nitrogen concentrations when compared to the observed concentrations. The second similarity was that both this project and the Yoon study simulations estimated the phosphorus leachate concentrations to be very low and almost constant over time. The observed phosphorus leachate data from the Yoon study and this project had more fluctuations than the simulated concentrations. Thus, the GLEAMS project estimations exhibited similar trends to the estimations from the study by Yoon et al. (1994).

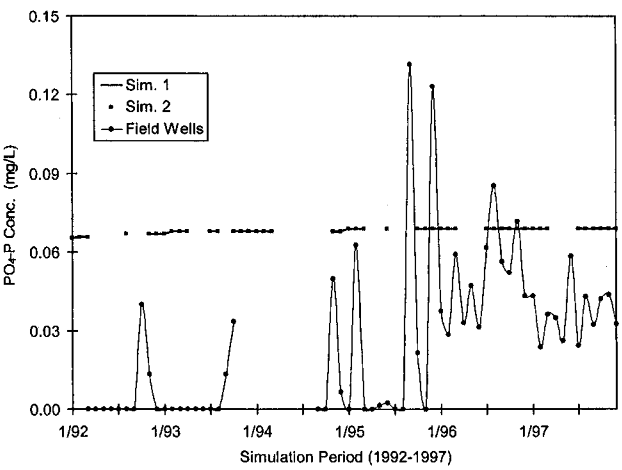


Figure 5. Observed and GLEAMS simulated PO_4–P leachate concentrations during the study period.

Table 5. Average monthly PO_4–P concentrations.

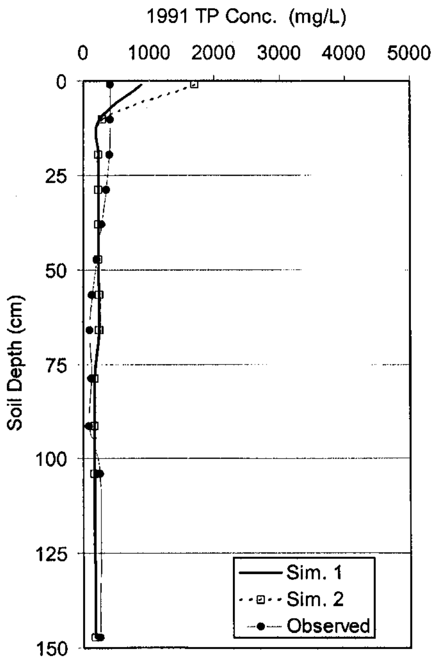
	Y_A (mg/L)	%Δ
GLEAMS Sim. 1	0.043	98.8
GLEAMS Sim. 2	0.043	98.4
Field wells	0.021	

GLEAMS Limitations

GLEAMS had some limitations. First, the estimation of phosphorus in the soil and the leachate was poor. There was little difference in the phosphorus concentrations between the simulations. With increased application rate, the phosphorus concentrations within the soil and leachate should have increased over time. All of the peak PO_4–P concentrations in the leachate were the same, regardless of simulation or time. In conjunction with this, the TP concentrations in the soil at depths below 20 cm did not differ between simulations. Some differences were expected between simulation 1 and simulation 2 due to the different application rates. The lack of differences in the phosphorus inputs caused REMM to be less accurate in its estimation of the phosphorus concentrations within the riparian buffer zone.

REMM Simulations

The comparison of the observed and the simulated NO_3–N leachate concentrations over time resulted in a better estimation of the trends (fig. 7). The observed leachate data at the edge of the field and the edge of the stream were compared to the simulated leachate concentrations in zone 1. All of the simulations indicated a larger decrease of the total NO_3–N concentrations than the observed data. The decrease in the total NO_3–N concentrations in the observed leachate may have been reduced by flow from the stream entering the last zone. The observed NO_3–N data showed a significant decrease in concentration beginning in January 1994. This could be the result of the planting of the riparian trees in April 1993. The simulated NO_3–N did not show this trend because, for simplicity, the model assumed the trees were planted for the entire simulation period. The percent difference between



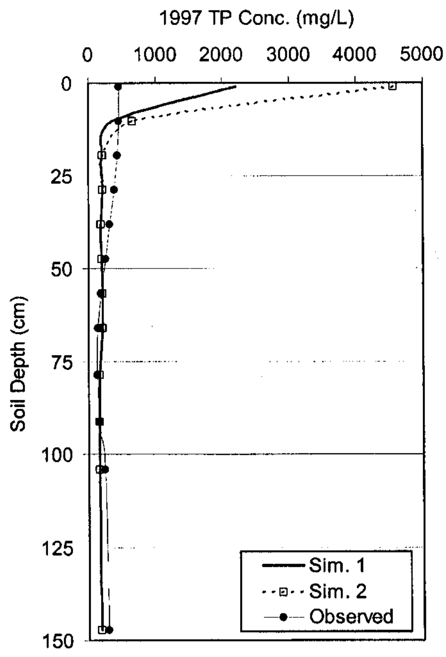


Figure 6. Observed and GLEAMS simulated TP soil concentrations versus soil depth for 1991 and 1997.

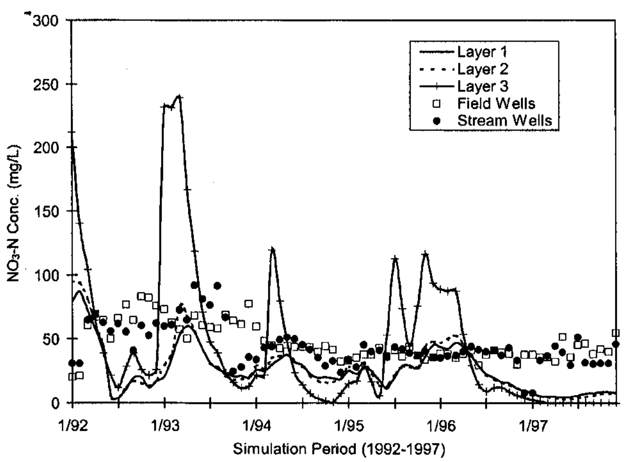


Figure 7. Observed and REMM simulated NO_3–N leachate concentrations for simulation 2 during the study period.

Table 6. Average monthly NO_3–N concentrations.

	Y_A (mg/L)	%Δ
REMM Sim. 1	0.45	–99.0
REMM Sim. 2	46.7	5.4
Stream wells	44.3	

the observed average NO_3–N concentration and simulation 1 was 99% (table 6). The percent difference between the observed average NO_3–N concentration and simulation 2 was 5%. This indicated that the actual application rate was likely even closer to 1000 kg N ha^{-1} yr^{-1} than GLEAMS had estimated.

The simulated dissolved labile phosphorus (LP) concentrations were compared to the PO_4–P concentrations measured in the well samples (fig. 8). The LP concentrations of the two simulations were almost identical for each layer. Both simulations overestimated the observed average monthly PO_4–P concentration. The LP concentrations were fairly constant over time with only minor fluctuations. The average monthly concentration in layer 3 was compared to the model's monthly concentrations (table 7). The model overestimated the phosphorus concentrations. However, there was a general agreement concerning the low range of the phosphorus concentrations.

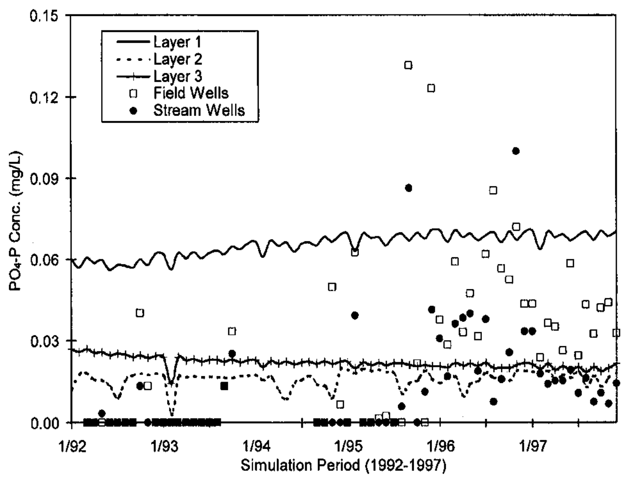


Figure 8. Observed PO_4–P and REMM simulated LP leachate concentrations for simulation 2 during the study period.

Table 7. Average monthly PO_4–P/LP concentrations.

	Y_A (mg/L)	%Δ
REMM Sim. 1	0.022	62.0
REMM Sim. 2	0.022	60.2
Stream wells	0.014	

REMM Limitations

Although REMM is capable of simulating multiple vegetation types per zone, only one vegetation type per zone was suggested since multiple vegetation types had not yet been tested. This may limit the accuracy of the output because the project site had multiple vegetation types in zones 1 and 2. In addition, only three soil layers per zone and a litter layer were available, which generalized the soil profile characteristic. A soil input structure similar to that used in GLEAMS would provide a better relationship between the two models, although this is not essential to coupling the two models. In addition, it was not practical to obtain all the required input data. This pertained primarily to various forms of carbon, nitrogen, and phosphorus in the soil and vegetation. The soil samples were not typically analyzed for these additional forms of nutrients. These limitations are being considered for improvements to future versions of REMM (Lowrance, 1999, personal communication).

Conclusions and Recommendations

The objective of this project was to use GLEAMS in combination with REMM to estimate the transport of nutrients through a riparian buffer zone from an agricultural field that received swine lagoon effluent. GLEAMS estimations of the nutrient concentrations leaving the spray field were first evaluated. The results of the evaluation indicated GLEAMS provided a reasonable estimation of the average nitrogen concentrations leaving the field. The simulated phosphorus leachate output revealed a potential limitation of GLEAMS' ability to estimate the phosphorus concentration differences between the two simulations. The limitations of the phosphorus estimations affected REMM's ability to accurately estimate the phosphorus movement within the riparian buffer zone.

REMM's ability to estimate the nutrient transport through the riparian buffer zone was also evaluated. The model was able to effectively estimate the NO_3–N leachate concentrations. The estimations of the phosphorus movement were limited by the GLEAMS' input phosphorus data. REMM's estimations were partially limited by the inability to acquire more real input data.

Recommendations for improving the REMM model were suggested and include the use of more soil layers to better describe the soil profile within each zone, additional testing using multiple vegetation types per zone, and improvements to the model's output structure. With these improvements, the REMM model would be a useful tool in the development of agricultural management plans for pollution reduction.

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